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ARTICLE

Effects of Marsh Terracing on Nekton Abundance at Two Locations in Galveston Bay, Texas

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Abstract We evaluated two marsh terracing restoration projects (GI=Galveston Island State Park, PM=Pierce Marsh) to compare nekton density and biomass between locations and among habitat types (open water pre-construction, open water post-construction, terrace marsh, terrace pond). Most (72%) animals collected were decapod crustaceans. Few differences in nekton abundance were observed over time (pre-construction versus post-construction open water). Comparisons of pre-construction to post-construction open water showed that the locations differed environmentally (e.g., turbidity) and in populations of nekton species. Density and biomass of gulf menhaden *Brevoortia patronus* and spot *Leiostomus xanthurus* in spring and density of white shrimp *Litopenaeus setiferus* in fall were higher at PM than GI, whereas densities and biomass of blue crab *Callinectes sapidus*, density of pink shrimp *Farfantepenaeus duorarum*, and biomass of brown shrimp *Farfantepenaeus aztecus* in the fall were higher at GI than PM. Despite location differences, constructing terraces appeared to benefit fishery species at both locations. Densities of brown shrimp, blue crab, white shrimp (fall), and pink shrimp (fall) and the biomass of brown shrimp, blue crab, and white shrimp (fall) were higher in terrace marsh than open water. Marsh terracing appears to be an effective tool for enhancing fishery habitat in degraded coastal systems.

Keywords Decapods · Fishes · Gulf of Mexico · Habitat value · Restoration

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Introduction

Marshes are often constructed to replace fishery habitat in coastal areas that have experienced marsh losses (Minello and Webb 1997; Thom et al. 2004; Rozas and Minello 2007; Rozas et al. 2007). Coastal marshes provide a nursery area for nekton (McIvor and Odum 1988; Moksnes et al. 1997; Beck et al. 2001; Minello et al. 2003). Marsh edge habitat, marsh within a few meters of the marsh-water interface, is a particularly important habitat for fishery species (Peterson and Turner 1994; Howe et al. 1999; Minello and Rozas 2002). The continued loss of marsh and other coastal habitats may have long-term negative effects on fisheries sustainability (Jordan et al. 2009).

Losses of marsh have been extensive in coastal Texas and Louisiana in the recent past (White et al. 1993; Barras et al. 2003), and some of these losses have been offset using marsh terracing, a restoration technique that increases marsh edge effectively and relatively inexpensively (Underwood et al. 1991). Marsh terraces are constructed by excavating sediment from shallow nonvegetated bottom and using this sediment to form ridges of emergent land adjacent to the borrow areas (Turner and Streever 2002). Constructed terraces can vary by configuration, side slope, and crown width. Regardless, this restoration technique is intended to convert areas of shallow open water to a mix of marsh and open water and thereby increase marsh area and marsh edge habitat.

Two terracing projects in Galveston Bay, Texas were constructed June to October 1999 at Galveston Island State Park (GI) and Pierce Marsh (PM). These projects were funded to increase fishery habitat as part of the settlement terms for the Apex Galveston oil spill in the Galveston Bay system that occurred in 1990 (Texas General Land Office et al. 1997). Before these projects were constructed, marsh edge at GI had decreased by 70% between 1982 and 1995 (Rozas et al. 2007) and we suspect similar losses have occurred at PM. Losses of marsh in these areas of Galveston Bay are generally the result of subsidence and erosion caused by withdrawal of subsurface water, oil, and gas (White and Tremblay 1995). Terraces were constructed in a checkerboard pattern with open corners that permit water flow and organism transit (Fig. 1). Construction specifications for both terrace projects, such as width and side slope, were based on those built in Louisiana and described by Underwood et al. (1991). Terraces at GI were constructed with various cell sizes as seen in Fig. 1, whereas the terraces at PM consisted of 60 m by 60 m cells only. The terraces were planted with marsh vegetation to retain the newly placed sediment and promote marsh development. Some terrace ponds at GI were planted with mixed seagrasses in 2000, but subsequent monitoring of these ponds showed that these plants did not survive.

Marsh terracing increasingly has been used for habitat restoration (Underwood et al. 1991), but the capability of the method to create a functional habitat has not been widely documented. Although both density and biomass of fishery species were greater in a nine-year old terrace marsh than over nearby shallow nonvegetated bottom, Rozas and Minello (2001) concluded that this terrace marsh was not functionally equivalent to natural marsh for some species. Pre-construction data, however, were not available for their study, and the study was limited to only one terrace site located in a mesohaline marsh of southwest Louisiana. Similarly, La Peyre et al. (2007) concluded that the density of fishery species was greater in terrace ponds than in shallow open water. Pre-construction data was unavailable for their study. In another study, Thom et al. (2004) concluded that nekton abundance increased as a result of terrace construction, but some species in shallow water were less abundant near terraces (terrace pond) than near natural marsh (natural pond) or sites treated with coconut mats (a sediment stabilization method). Rozas et al. (2005) examined different types of restoration techniques using a modeling approach and found marsh terracing to be the most cost-effective method that increased populations of fishery species. Initial site conditions affected project costs in their study, but the influence of those site differences on the effectiveness of fishery habitat restoration is unknown.

The objective of our study was to evaluate marsh terracing as a method for restoring estuarine habitat in a polyhaline region of Galveston Bay. Two marsh terracing projects were constructed at roughly the same time, but differed in the total area of restoration, physical location, and environmental conditions. The differences between these projects and their locations provided an opportunity to determine whether such differences influence the use of terrace habitat by nekton 2 to 3 years after project construction. We compared nekton density and biomass between two locations (GI and PM) and among nekton habitats (terrace marsh, terrace pond, and open water references) in lower Galveston Bay. Our study is unique in comparing pre-construction and post-construction nekton use of shallow open water at the two locations. This open water habitat is shallow nonvegetated bay bottom, some of which was replaced by terrace construction. Differences in nekton use of this open water habitat before and after terrace construction are attributed to temporal changes in nekton abundance or impacts of nearby terrace field construction. Both pre-construction and post-construction open water are references for the marsh terracing habitats (terrace pond and terrace marsh).

Methods

Our study area included marsh terracing projects at GI and PM. These restoration projects were located in polyhaline estuary along the upper Texas coast (Fig. 1). Tides are predominantly diurnal with a mean daily range of 0.3 m (Orlando et al. 1991). The vegetation of marshes in the study area was predominately smooth cordgrass *Spartina alterniflora*. The GI terraces were located in Carancahua Cove on the bay side of Galveston Island, where decreasing fishery populations have coincided with wetland loss (Rozas et al. 2007). The PM restoration area was located approximately 9.7 km north of GI. The project area (total terrace area + a 25 m wide buffer) at PM was 0.316 km^2 and contained 17.7% marsh (Rozas et al. 2005). The project area at GI was nearly twice that at PM (0.682 km^2), but with a similar percentage of marsh (18.6%).

Our study was designed to examine whether habitat created by terracing supported higher densities of fishery and foraging organisms than the shallow open water replaced by the restoration projects. For the purpose of this study, we included data from only 60 m by 60 m terrace cells at GI to match the size of terrace cells at PM. We excluded smaller (30 m by 30 m) and larger (122 m by 122 m) terrace cells at GI in our study, but all three sizes were included in an earlier study that examined the effect of

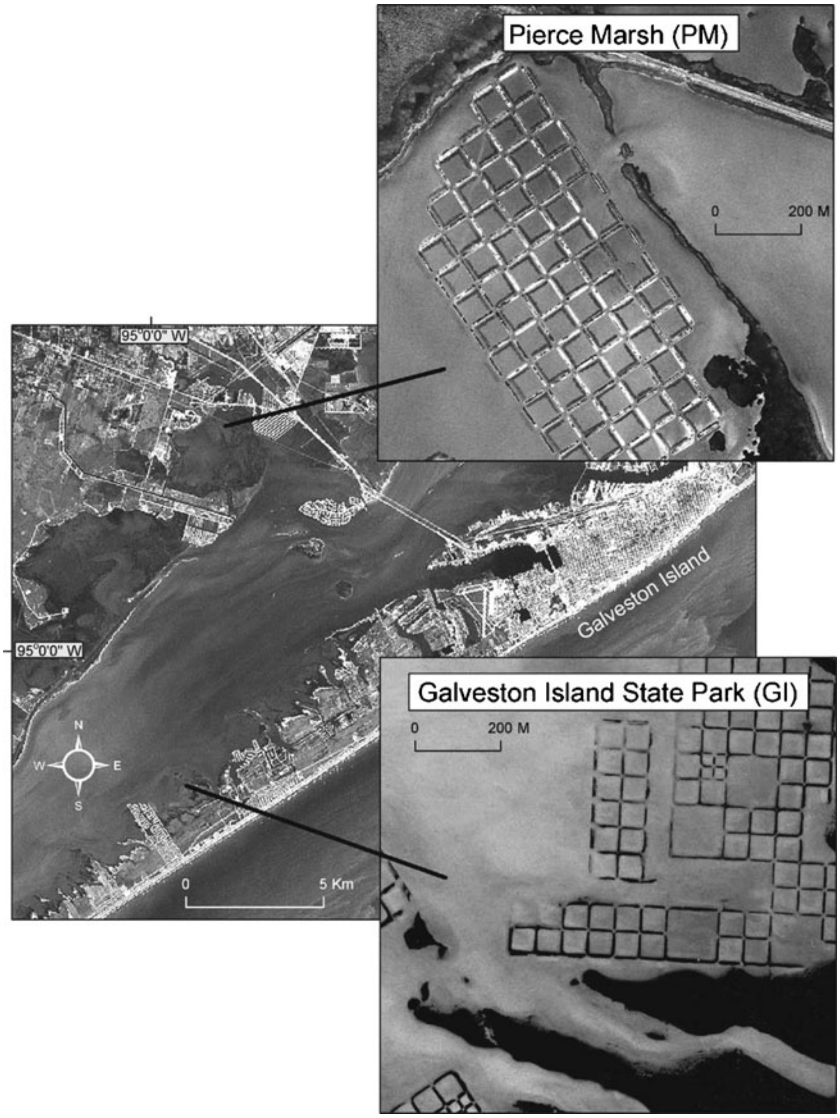

Fig. 1 Aerial photograph of the study area with marsh terraces shown at the two locations of Galveston Island State Park (GI) and Pierce Marsh (PM). The study area is located within the Galveston Bay estuary

terrace cell size on habitat use (Rozas and Minello 2007). We identified four habitat types to sample: terrace marsh (marsh constructed by terracing), terrace pond (semi-enclosed water within the terrace cells), pre-construction reference area (PREREF, open water where terraces were to be constructed), and post-construction reference area (POSTREF, open water outside the terraces). Each sample site was selected randomly and identified in the field using aerial photography.

Nekton was sampled with a 1-m^2 drop sampler following Zimmerman et al. (1984). Water temperature, dissolved oxygen (DO), salinity, turbidity, depth, and number of marsh stems (smooth cordgrass) were recorded using the methods described by Minello and Zimmerman (1992) and Rozas and Minello (2007). Distance from the center of the sample to the nearest marsh was recorded as "distance to edge". Ten PREREF samples were collected at each location during May and October 1998 before the terraces were constructed, and seven samples each were collected following terrace construction from the three habitat types (terrace marsh, terrace pond, POSTREF) at each location during September 2001 and May 2002. Samples were collected when high densities of fishery species are known to inhabit estuaries of the northern Gulf of Mexico (Rakocinski et al. 1992; Livingston 1997; Akin et al. 2003; Rozas et al. 2007). Animals were removed from the drop sampler with dip nets or by hand after removing water from the drop sampler, and preserved in formalin until processed in the laboratory.

In the laboratory, samples were sorted to remove animals, and these animals were identified to lowest feasible taxon using Heard (1982), Williams (1984), Abele and Kim (1986), Hoese and Moore (1998), McEachran and Fechhelm (1998), and Richards (2005). Wet biomass was recorded for each taxon in a sample. Organisms that could not be identified to species were assigned to species based on the proportion (density and biomass) of identified species occurring in each sample.

We used a two-way Multivariate Analysis of Variance (MANOVA) to examine the null hypotheses that density and biomass of the crustacean species did not differ by location or habitat (Proc GLM, SAS version 9.1, SAS Institute Inc. Cary, NC, USA). We conducted a similar but separate analysis for fish species, assuming that these two groups (decapod crustaceans and fish) are independent of one another. To account for expected seasonal variation in nekton assemblages, the data from the two groups were analyzed separately by seasons, spring and fall. Using what Scheiner and Gurevitch (2001) describe as a hybrid analysis, we used Analysis of Variance (ANOVA) for selected species after verifying significance in MANOVA results (SuperAnova v1.11 by Abacus Concepts, Berkeley, CA). Species selected for analysis included any contributing >1% of the total number or >10% of the total biomass for fishes or crustaceans within either spring or fall (Table 1 and 2). We used *a priori* contrasts to determine differences in habitat when the Habitat main effect was significant. We contrasted 1) PREREF vs. POSTREF, 2) POSTREF vs. terrace pond, and 3) POSTREF vs. terrace marsh. The first contrast compared nekton use of open water before and after terrace construction to examine temporal change, while the other two contrasts compared the constructed habitat types (terrace pond and terrace marsh) to a reference (POSTREF), the habitat type replaced by marsh terracing. We considered results significant at $p<0.05$, but also assessed significance of univariate analyses after adjusting alpha levels for the main effects with the Bonferroni method described by Rice (1989), which accounts for errors that can result from making multiple comparisons. We also conducted *a posteriori* Games-Howell tests to evaluate all pairs in the ANOVA.

Transformations of some data were required prior to analysis to meet MANOVA and ANOVA assumptions. Densities and biomass of animals were not normally distributed. Therefore, these data were transformed using the natural log prior to analysis. Variances of environmental variables were not homogeneous. Natural log transformations corrected variances for depth, turbidity and temperature; but log, square, and square root transformations failed to correct variances for salinity and DO distribution. Therefore, environmental data were analyzed with a randomization method described by Adams and Anthony (1996). This method compares the observed treatment sum-of-squares (*SS*) for the data to *SS* generated from observations randomly assigned to treatments. The frequency of observations in a treatment is maintained, the observed data are randomly assigned to treatments, and *SS* is calculated. The random assignment of observations to treatments and calculation of the *SS* is repeated 5,000 times. The frequency of randomly generated *SS* greater than or equal to the observed *SS* is an estimate of the likelihood of obtaining the observed *SS* by chance. Therefore, data may be analyzed without any assumptions of distribution. An alpha level of 0.05 was used to reject the null hypotheses. We present untransformed data throughout this paper, after verifying transformed and untransformed data display similar trends.

Results

We collected 1,298 fishes and decapod crustaceans in spring with a combined weight of 1,066.8 g. In fall, we collected 1,106 (465.7 g) fishes and decapod crustaceans. Decapod crustaceans were more abundant than fishes, comprising 72% of the 2,404 organisms collected. Daggerblade grass shrimp *Palaemonetes pugio*, brown shrimp *Farfantepenaeus aztecus*, white shrimp *Litopenaeus setiferus*, blue crab *Callinectes sapidus* and brackish grass shrimp *Palaemonetes intermedius* accounted for 94% of all crustaceans, whereas 45% of all fishes were gulf menhaden *Brevoortia patronus*. Gulf menhaden accounted for 66% of all fish collected in the spring, but did not occur in the fall. The most abundant fishes in the fall were bay anchovy *Anchoa mitchili* (33%), darter goby *Ctenogobius bolesosoma* (18%), clown goby *Microgobius gulosus* (14%), and naked goby *Gobiosoma bosc* (9%). Other taxa infrequently collected in our study included bay whiff *Citharichthys spilopterus*, blackcheek tonguefish *Symphurus plagiusa*, code goby *Gobiosoma robustum*, marsh grass shrimp *Palaemonetes vulgaris*, gulf killifish *Fundulus grandis*, lined sole *Achirus lineatus*, mojarras *Gerreidae sp*., pinfish *Lagodon rhomboides*, least puffer *Sphoeoroides parvus*, sand seatrout *Cynoscion arenarius*, silver perch *Bairdiella chrysoura*, inland silverside *Menidia berylina*, bigclaw snapping shrimp *Alpheus heterochaelis*, southern flounder *Paralichtys lethostigma*, spot *Leistomus xanthurus*, spotted seatrout *Cynoscion nebulosus*, striped mullet *Mugil cephalus*, and speckled worm eel *Myrophis punctatus*.

In multivariate analysis, the overall location by habitat interaction was significant for decapod crustaceans both spring and fall ($p<0.001$), and fish in spring ($p<0.0001$), and for fish in the fall ($p=0.004$). The main effects of location and habitat were significant for both decapod crustaceans and fishes of both seasons ($p<0.003$). Univariate and multivariate analysis provided similar results; we provide the univariate means comparisons for the significant effects below.

Pre-construction Open Water Reference Versus Post-Construction Open Water Reference

Relatively few differences in density or biomass were detected between PREREF and POSTREF samples in our

Table 1 Comparison of mean densities (m^{-2}) and standard error (S.E.) of the most abundant decapod crustaceans and fishes collected at two locations and four habitat types (PREREF=1998 pre-construction open water, POSTREF=2001–2002 post-construction open water, terrace pond, and terrace marsh). Species richness was determined by enumerating the number of species in each sample. The probabilities (p-values) are given for two-way ANOVAs used to compare mean densities among the different levels in each main effect on natural log transformed data. We used *a priori* contrasts within the Habitat Type main effect to test for significant differences between: C-1=PREREF and POSTREF, C-2=POSTREF and terrace pond, and C-3=POSTREF and terrace marsh. The results (p-values) also are given for tests of the L x H=Location by Habitat Type interaction. An * indicates that the probability value was significant after alpha was adjusted as described by Rice (1989)

Species	Location main effect					Habitat type main effect									Contrast p-values			L × H
	Galveston Island		Pierce marsh		ANOVA	PREREF		POSTREF		Terrace pond		Terrace Marsh		ANOVA				
	n=31		*n*=31			*n*=20		*n*=14		*n*=14		*n*=14						
	Mean	S. E.	Mean	S. E.	*p*-value	Mean	S. E.	Mean	S. E.	Mean	S. E.	Mean	S. E.	*p*-value	C-1	C-2	C-3	*p*-value
Spring																		
Total crustaceans (9 spp)	9.58	3.63	17.10	5.77		0.95	0.22	5.71	4.22	1.71	0.46	50.29	9.36					
Palaemonetes pugio	3.84	1.88	8.16	2.98	0.1003	0.00	0.00	3.57	3.42	0.00	0.00	23.00	4.89	0.0001*	0.1508	0.1847	0.0001	0.3124
Farfantepenaeus aztecus	4.58	1.50	3.48	0.97	0.4104	0.60	0.20	1.79	0.74	1.64	0.48	13.57	2.52	0.0001*	0.13	0.7693	0.0001	0.0203
Palaemonetes intermedius	0.19	0.19	4.26	1.99	0.0001*	0.00	0.00	0.00	0.00	0.00	0.00	9.86	4.01	0.0001*	1	1	0.0001	0.0001*
Callinectes sapidus	0.65	0.28	0.71	0.26	0.6567	0.00	0.00	0.29	0.22	0.07	0.07	2.64	0.56	0.0001*	0.3205	0.5403	0.0001	0.6386
Farfantepenaeus duorarum	0.03	0.03	0.03	0.03	1	0.10	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.2664				1
Litopenaeus setiferus	0.10	0.07	0.00	0.00	0.1583	0.15	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.2434				0.2434
Total fishes (23 spp)	4.42	1.91	10.77	3.58		9.55	3.56	1.21	0.93	11.93	7.29	6.86	1.43					
Brevoortia patronus	1.65	1.65	8.13	3.45	0.0014*	7.55	3.10	0.57	0.57	10.29	7.19	0.00	0.00	0.0074*	0.0079	0.1525	0.6672	0.0046*
Lagodon rhomboides	1.90	0.70	0.81	0.25	0.2241	0.40	0.15	0.14	0.14	0.07	0.07	5.21	1.16	0.0001*	0.2055	0.8413	0.0001	0.0001*
Gobiosoma bosc	0.45	0.22	0.19	0.16	0.2276	0.00	0.00	0.00	0.00	0.64	0.39	0.79	0.43	0.0286	1	0.0624	0.0263	0.1368
Leiostomus xanthurus	0.03	0.03	0.52	0.15	0.0001*	0.75	0.20	0.07	0.07	0.07	0.07	0.00	0.00	0.0001*	0.0001	1	0.4248	0.0001*
Achirus lineatus	0.00	0.00	0.23	0.17	0.1011	0.35	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.1328				0.1328
Microgobius gulosus	0.13	0.13	0.03	0.03	0.6003	0.00	0.00	0.00	0.00	0.36	0.29	0.00	0.00	0.1243				0.8126
Species richness (count)	1.97	0.39	3.58	0.38	0.0001*	2.40	0.44	1.50	0.55	1.86	0.33	5.50	0.37	0.0001*	0.0787	0.0942	0.0001	0.0046*
Fall																		
Total crustaceans (9 spp)	17.81	5.38	11.13	2.98		1.15	0.34	1.36	0.34	6.00	1.06	55.07	5.52					
Litopenaeus setiferus	5.26	1.85	5.48	1.41	0.0218	0.50	0.20	0.21	0.16	2.86	0.94	20.00	2.27	0.0001*	0.3159	0.0001	0.0001	0.0001*
Palaemonetes pugio	3.81	1.64	2.77	1.05	0.765	0.00	0.00	0.07	0.07	0.00	0.00	14.50	2.63	0.0001*	0.6771	0.701	0.0001	0.975
Callinectes sapidus	4.07	1.10	0.77	0.27	0.0001*	0.15	0.15	0.36	0.17	1.57	0.44	8.57	1.84	0.0001*	0.2837	0.0018	0.0001	0.0001*
Farfantepenaeus aztecus	3.13	1.05	1.45	0.51	0.3905	0.45	0.17	0.57	0.23	0.21	0.16	8.71	1.71	0.0001*	0.6444	0.1828	0.0001	0.0001*
Farfantepenaeus duorarum	1.52	0.47	0.29	0.12	0.0022*	0.00	0.00	0.07	0.07	1.29	0.55	2.64	0.80	0.0001*	0.7261	0.0011	0.0001	0.003*
Palaemonetes intermedius	0.32	0.32	0.32	0.32	1	0.05	0.05	0.00	0.00	0.00	0.00	0.07	0.07	0.6287				0.3029
Total fishes (23 spp)	3.45	0.64	3.29	0.73		1.10	0.36	4.29	1.17	4.21	1.22	4.86	0.92					
Anchoa mitchili	1.32	0.48	0.94	0.51	0.3292	0.65	0.28	1.79	1.06	2.29	0.98	0.00	0.00	0.0284	0.2923	0.5661	0.0236	0.0864
Ctenogobius boleosoma	0.00	0.00	1.48	0.45	0.0001*	0.00	0.00	0.64	0.27	0.50	0.31	2.14	0.91	0.0008*	0.01	0.4414	0.0212	0.0008*
Microgobius gulosus	0.84	0.21	0.13	0.08	0.0001*	0.00	0.00	1.00	0.35	1.00	0.31	0.14	0.10	0.0001*	0.0001	0.9189	0.0017	0.0035*
Gobiosoma bosc	0.45	0.24	0.13	0.13	0.1673	0.20	0.20	0.36	0.29	0.07	0.07	0.57	0.44	0.6583				0.1523
Leiostomus xanthurus	0.16	0.07	0.03	0.03	0.0922	0.10	0.07	0.21	0.11	0.07	0.07	0.00	0.00	0.2952				0.8096
Lagodon rhomboides	0.13	0.08	0.00	0.00	0.0758	0.00	0.00	0.07	0.07	0.00	0.00	0.21	0.16	0.1582				0.1582
Species richness (count)	3.74	0.46	3.36	0.47	0.2615	1.40	0.29	3.00	0.54	3.64	0.44	7.07	0.29	0.0001*	0.0005	0.023	0.0001	0.2578

Table 2 Comparison of mean biomass (g m^{-2}) and standard error (S.E.) of decapod crustaceans and fishes contributing the most biomass in samples collected at two locations and four habitat types (PREREF=1998 pre-construction open water, POSTREF=2001–2002 post-construction open water, terrace pond, and terrace marsh). The probabilities (p-values) are given for two-way ANOVAs used to compare mean biomass among the different levels in each main effect on natural log transformed data. We used *a priori* contrasts within the Habitat Type main effect to test for significant differences between: C-1=PREREF and POSTREF, C-2= POSTREF and terrace pond, and C-3=POSTREF and terrace marsh. The results (p-values) also are given for tests of the Location by Habitat Type Interaction (L x H). An * indicates that the probability value was significant after alpha was adjusted as described by Rice (1989)

Species	Location main effect					Habitat type main effect									Contrast p-values			L × H
	Galveston Island		Pierce marsh		ANOVA	PREFER		POSTREF		Terrace pond		Terrace marsh		ANOVA				
	n=31		*n*=31			*n*=20		*n*=14		*n*=14		*n*=14						
	Mean	S. E.	Mean	S. E.	*p*-value	Mean	S. E.	Mean	S. E.	Mean	S. E.	Mean	S. E.	*p*-value	C-1	C-2	C-3	*p*-value
Spring																		
Total crustaceans (9 spp)	9.6	4.8	6.3	2.0		0.9	0.3	2.0	1.1	1.2	0.4	30.7	9.3					
Callinectes sapidus	4.4	3.6	2.1	1.1	0.8052	0.0	0.0	0.1	0.1	0.0	0.0	14.4	7.8	0.0001*	0.7935	0.8102	0.0001	0.9931
Farfantepenaeus aztecus	3.9	1.2	1.8	0.4	0.6393	0.6	0.3	1.2	0.6	1.2	0.4	9.2	2.1	0.0001*	0.3589	0.607	0.0001	0.0003*
Palaemonetes pugio	1.0	0.5	1.5	0.5	0.1884	0.0	0.0	0.6	0.6	0.0	0.0	4.7	1.1	0.0001*	0.353	0.391	0.0001	0.528
Litopenaeus setiferus	0.0	0.0	0.0	0.0	0.2028	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.333				0.333
Total fishes (23 spp)	8.1	5.6	10.5	4.1		6.8	2.3	0.8	0.6	13.9	8.6	16.7	12.3					
Paralichthys lethostigma	5.4	5.4	0.0	0.0	0.3365	0.0	0.0	0.0	0.0	0.0	0.0	1.9	11.9	0.3431				0.3238
Brevoortia patronus	0.4	0.4	4.5	2.1	0.0012*	3.8	1.7	0.4	0.4	5.0	4.1	0.0	0.0	0.0122*	0.0121	0.1814	0.6574	0.01
Lagodon rhomboides	1.8	0.6	0.7	0.2	0.1606	0.5	0.2	0.2	0.2	0.3	0.3	4.2	1.1	0.0001*	0.2429	0.8674	0.0001	0.0004*
Mugil cephalus	0.0	0.0	3.7	3.7	0.2799	0.0	0.0	0.0	0.0	8.1	8.1	0.0	0.0	0.356				0.356
Leiostomus xanthurus	0.1	0.1	1.1	0.5	0.0009*	1.5	0.8	0.2	0.2	0.2	0.2	0.0	0.0	0.0004*	0.0018	0.9368	0.4975	0.0001*
Fall																		
Total crustaceans (9 spp)	4.8	1.7	1.4	2.1		0.3	0.2	0.1	0.0	0.8	0.3	12.3	2.8					
Litopenaeus setiferus	2.1	0.8	0.8	0.2	0.0878	0.3	0.2	0.0	0.0	0.7	0.3	5.3	1.4	0.0001*	0.2466	0.0027	0.0001	0.0001*
Farfantepenaeus aztecus	1.1	0.4	0.1	0.0	0.0001*	0.0	0.0	0.1	0.0	0.0	0.0	2.5	0.8	0.0001*	0.8757	0.5882	0.0001	0.0001*
Callinectes sapidus	0.7	0.3	0.2	0.1	0.0006*	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.5	0.0001*	0.9949	0.8574	0.0001	0.0001*
Palaemonetes pugio	0.6	0.3	0.3	0.1	0.3676	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.5	0.0001*	0.9923	0.9929	0.0001	0.4492
Total fishes (23 spp)	4.8	1.6	4.0	2.4		3.4	2.2	3.4	1.5	4.9	3.9	6.4	3.8					
Leiostomus xanthurus	2.9	1.5	0.2	0.2	0.067	3.2	2.2	1.6	0.9	0.7	0.7	0.0	0.0	0.4391				0.6669
Mugil cephalus	0.0	0.0	2.4	1.8	0.1638	0.0	0.0	0.0	0.0	3.9	3.9	1.4	1.4	0.4801				0.4801
Lagodon rhomboides	1.3	0.8	0.0	0.0	0.08	0.0	0.0	1.4	1.4	0.0	0.0	1.3	1.0	0.2609				0.2609
Paralichthys lethostigma	0.0	0.0	1.0	1.0	0.3179	0.0	0.0	0.0	0.0	0.0	0.0	2.2	2.2	0.333				0.333

analysis, and these differences were not consistent between spring and fall (Tables 1 and 2). In spring, two pelagic species, gulf menhaden and spot were more abundant (and biomass was higher) in open water habitat before than after terrace construction, whereas in fall, species richness and densities of darter goby and clown goby were higher in open water following terrace construction (Tables 1 and 2). Neither density nor biomass of decapod crustaceans in open water differed significantly between PREREF and POSTREF samples (Tables 1 and 2).

Galveston Island Versus Pierce Marsh

Density or biomass of several taxa differed by location (Tables 1 and 2). Densities of brackish grass shrimp in spring and white shrimp in fall were greater at PM than GI, whereas blue crab and pink shrimp in fall were more abundant at GI than PM. Species richness and densities of gulf menhaden, spot in spring, and darter goby in fall were higher at PM than GI, whereas clown goby (fall) was the only fish more abundant at GI than PM. Comparisons of biomass between locations revealed a similar pattern, but fewer significant differences were detected in biomass than density (Tables 1 and 2). Biomasses of gulf menhaden and spot in spring were greater at PM than GI, whereas those of brown shrimp and blue crab in fall were greater at GI than PM (Table 2).

Habitat Type Effects

Densities and biomasses of most decapod crustacean species differed significantly among habitat types (Tables 1 and 2). Brown shrimp, blue crab, white shrimp (fall), pink shrimp (fall), daggerblade grass shrimp, and brackish grass shrimp (spring) were all more abundant in terrace marsh than at POSTREF sites. Biomasses of brown shrimp, blue crab, white shrimp (fall), and daggerblade grass shrimp were more abundant at terrace marsh than POSTREF sites as well (Table 2). In fall, densities of white shrimp, blue crab, and pink shrimp and biomass of white shrimp also were higher in terrace ponds than at POSTREF sites. Species richness was significantly higher in both terrace marsh and terrace ponds than at POSTREF sites in fall.

Fish density and biomass also differed among habitat types, but the pattern was less consistent. Pinfish and naked goby in spring and darter goby in fall were more abundant in terrace marsh than at POSTREF sites, whereas bay anchovy and clown goby were more abundant at POSTREF sites than in terrace marsh during fall. Pinfish biomass was greater in terrace marsh than at POSTREF sites, whereas biomasses of gulf menhaden and spot were greater at POSTREF sites than in terrace marsh in the spring (Table 2).

There were significant Location by Habitat interactions with density for five species in spring and six species in fall, and with biomass for four species in the spring and three species in fall. Means of brown shrimp biomass and density in terrace marsh were higher at GI than PM, but higher at PM than GI at POSTREF (spring) and PREREF (fall) sites. Even so, these means for brown shrimp within each location were higher in terrace marsh than other habitat types (Fig. 2). In fall, blue crab biomass and density and pink shrimp density in terrace marsh and terrace ponds were higher at GI than PM, but similar between locations within the other habitat types. Within each location, however, both blue crab and pink shrimp were more abundant in terrace marsh than at POSTREF sites (Figs. 3 and 4). Blue crab also had more biomass in terrace marsh than at POSTREF sites. White shrimp density in fall at terrace pond sites was higher at GI than PM, but within other habitat types, similar between the two locations (Fig. 4). Biomass of white shrimp was also higher at GI than PM in terrace ponds in the fall. Although white shrimp was more abundant in terrace ponds than at POSTREF sites at GI, mean white shrimp densities in these two habitat

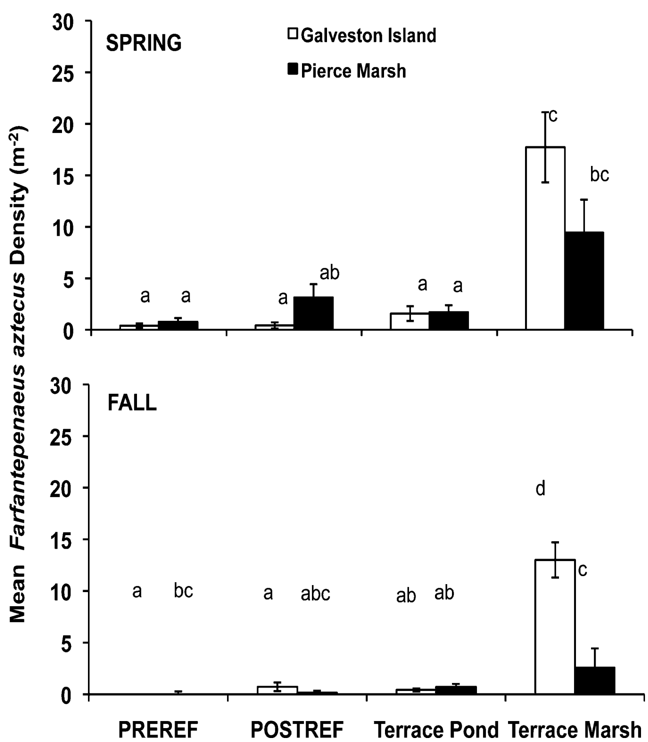

Fig. 2 Comparison of brown shrimp *Farfantepenaeus aztecus* density in spring (*above*) and fall (*below*) between Galveston Island State Park (GI) and Pierce Marsh (PM) and among four habitat types (preconstruction open water=PREREF, post-construction open water=POSTREF, terrace marsh, and terrace pond). *Error bars*=1 standard error. Mean densities were computed from 10 PREREF and seven POSTREF, terrace marsh, and terrace pond samples at each location (GI and PM). *Letters* represent significant differences of *a posteriori* Games-Howell tests on natural log transformations of the data presented

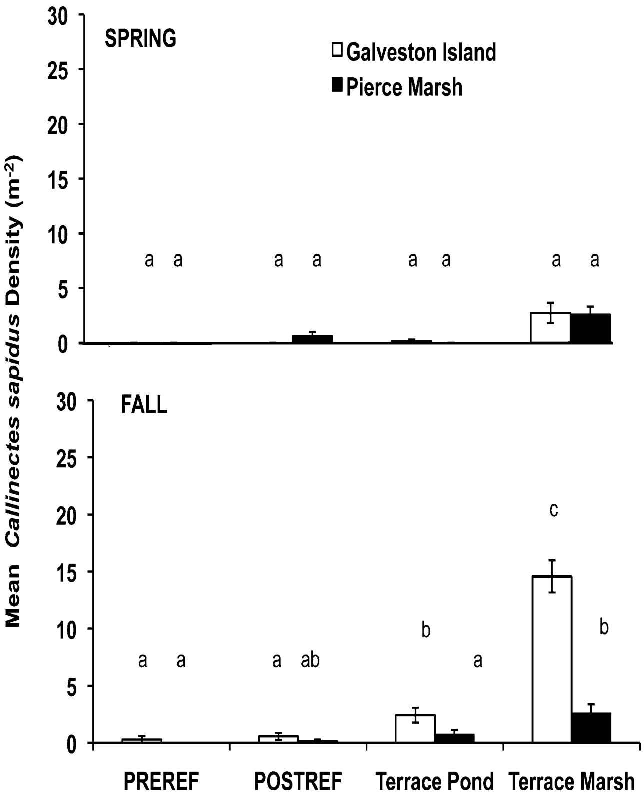

Fig. 3 Comparison of blue crab *Callinectes sapidus* density in spring (*above*) and fall (*below*) between Galveston Island State Park (GI) and Pierce Marsh (PM) and among four habitat types (pre-construction open water=PREREF, post-construction open water=POSTREF, terrace marsh, and terrace pond). *Error bars*=1 standard error. Mean densities were computed from 10 PREREF and seven POSTREF, terrace marsh, and terrace pond samples at each location (GI and PM). *Letters* represent significant differences of *a posteriori* Games-Howell tests on natural log transformations of the data presented

types were similar at PM (Fig. 4). In spring, mean pinfish densities at PREREF and POSTREF sites were higher at PM than GI, but higher at GI than PM at terrace pond and terrace marsh sites (Fig. 4). The biomass and density means for pinfish within each location also were higher in terrace marsh than at POSTREF sites. In spring, means of gulf menhaden and spot density (and biomass) at PREREF sites were higher at PM than GI, but similar between locations within the other habitat types (Fig. 4). No gulf menhaden or spot were collected in terrace marsh.

Environmental Variables

The PREREF and POSTREF sample sites differed in environmental characteristics in fall (Table 3). PREREF sites had higher salinity, higher turbidity, and lower DO than POSTREF sites in fall, but our analysis detected no difference in environmental variables between PREREF and POSTREF sites in spring.

Locations differed by environmental characteristics, but these differences were not consistent between spring and fall (Table 3). In spring, the GI sites were deeper, had higher salinity, and lower turbidity and water temperature than the PM sites. In fall, the GI sites had higher water temperature, but lower DO than PM sites. Sample sites in fall also were farther from the marsh at PM than GI.

Environmental variables also differed among habitat types after terrace construction (Table 3). In spring, terrace marsh sites had higher turbidity and water temperature than POSTREF sites. As expected terrace marsh sites were shallower than POSTREF sites.

The location by habitat type interaction was significant for three variables in analyses of the spring data and four variables of fall data (Table 3). Salinity measured in fall was higher at PREREF than POSTREF sites and differed little among habitat types sampled after the terraces were constructed. DO measured in spring did not differ between PREREF and POSTREF sites, but was lower in terrace ponds and higher at terrace marsh sites than at POSTREF sites.

Submerged aquatic vegetation (SAV) was scarce in the study area and rarely collected in our samples. The seagrass plantings at the GI location did not survive, but some SAV naturally occurred in our study area. *Ruppia maritima* was present in one PREREF sample at PM, and three POSTREF samples at GI contained *Halophila engelmanni* and *Halodule wrightii*.

Discussion

Constructing marsh terraces benefited fishery species at both GI and PM, despite existing differences in environmental characteristics and nekton populations. Brown shrimp density and biomass (averaged over spring and fall) were 8 to 13 times higher in terrace marsh than at POSTREF sites, the habitat replaced with terraces. Similarly, blue crab densities were 12 to 28 times greater, and their biomass 136 to 396 times more, in terrace marsh than at POSTREF sites in spring and fall averages. Although the nekton assemblage in our study area was predominantly crustaceans, fishes were more predominant than crustaceans in other studies (Thom et al. 2004; La Peyre et al. 2007). The results of these studies are consistent in finding that marsh constructed by terracing supports higher densities and more biomass of fishery and foraging species than the open water it replaces (Rozas and Minello 2001; Thom et al. 2004; La Peyre et al. 2007; Rozas and Minello 2007). Our study is important because it shows that terrace marsh can enhance fishery habitat within only 2 years after construction, and that terracing can be an effective restoration tool in the polyhaline environment of Galveston Bay.

The marsh terracing projects appeared to enhance nekton species diversity as well, which is often one of the goals of restoration. We found that species richness was statistically greater in terrace marsh and terrace ponds than in open

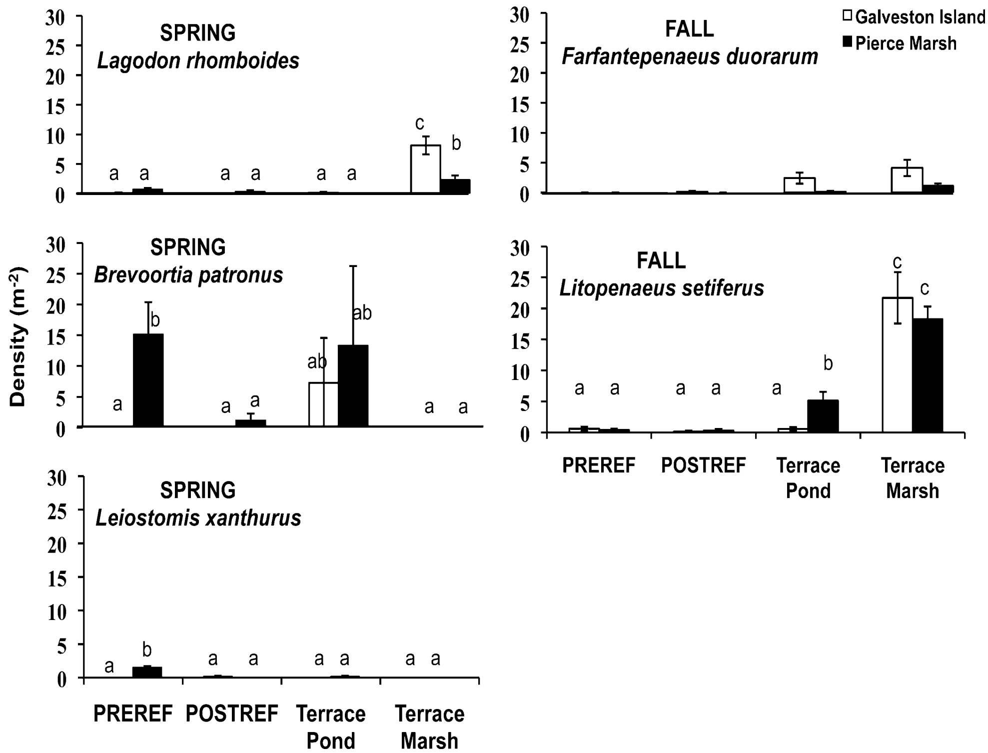

Fig. 4 Comparison of pinfish *Lagodon rhomboides*, gulf menhaden *Brevoortia patronus*, and spot *Leiostomis xanthurus* density in spring and pink shrimp *Farfantepenaeus duorarum*, white shrimp *Litopenaeus setiferus* density in fall between and among four habitat types (pre-construction open water=PREREF, post-construction open water=POSTREF, terrace marsh, and terrace pond). *Error bars*=1 standard error. Mean densities were computed from 10 PREREF and seven POSTREF, terrace marsh, and terrace pond samples at each location (GI and PM). *Letters* represent significant differences of *a posteriori* Games-Howell tests on natural log transformations of the data presented

water references. These results are similar to those of La Peyre et al. (2007), who reported that terrace edge had a greater number of species than open water but fewer than natural marsh edge. Minello and Webb (1997) did not detect a difference in species richness between natural and created marshes.

Studies based on density and biomass consistently show that marsh terracing appears to benefit some important fishery species, however, additional studies using other metrics are needed to fully assess this restoration technique (Rozas and Minello 2009). For example, La Peyre et al. (2007) used fish species composition and condition (weight:length) to assess the habitat quality of shallow ponds created by marsh terracing. They reported that terrace ponds lacked functional equivalency with natural marsh ponds based on differences in nekton assemblages between terrace and reference ponds and a poorer condition of nekton in terrace ponds for two of the six fish species tested.

The design of our study, which encompassed two different locations, enabled us to examine variability between locations. Replicating locations when assessing restoration methods should address concerns with pseudoreplication (Hurlberg 1984) and permit the application of study results beyond a single project. Although both locations in our study occur within the same bay system, are tidally connected, and both have experienced marsh loss in the recent past (White et al. 1993; White et al. 2004), there are environmental and ecological differences. Pierce Marsh is located near the mainland, exposed to wind and waves from the south, and may be more directly influenced by runoff from urban areas on the mainland than GI, which is located within a state park. The GI location is on a barrier island with exposure to wind and waves from the north. Soils at GI consist of sand to muddy sand, whereas soils at PM contain finer material (clay and silt). The fine sediment at PM and its close proximity to developed areas may explain the relatively high turbidity we observed there in the spring. The density and biomass of fishery species also differed between the two locations. Despite differences between locations, constructing marsh terraces increased marsh habitat and appeared to increase populations of penaeid shrimps and blue crab at both locations.

Although the terracing projects appeared to benefit penaeid shrimps and blue crab, the effect of these projects on other fishery species was variable. Gulf menhaden was relatively abundant in terrace ponds. Spot and gulf menhaden are typically associated with open water (Christmas et al. 1982; Stickney and Cuenco 1982) and seldom use marsh habitat directly (Minello 1999). Variable distribution is characteristic of gulf menhaden, a schooling species (Christmas et al. 1982). The low densities of spot and variable distribution of gulf menhaden we observed in our study make it difficult to draw conclusions about these species. Populations of darter and clown gobies may have increased in response to the increase in marsh habitat provided by the marsh terracing projects, although these

Table 3 Means and standard error (S.E.) for environmental characteristics collected at two locations and four habitat types (PREREF=1998 pre-construction open water, POSTREF=2001–2002 post-construction open water, terrace pond, and terrace marsh). The probabilities (p-values) are results of 5000 ANOVAs per variable with randomized data compared to actual sum of squares. We used *a priori* contrasts within the Habitat Type main effect to test for significant differences between: C-1=PREREF and POSTREF, C-2=POSTREF and terrace pond, and C-3=POSTREF and terrace marsh. The results (p-values) also are given for tests of the Location by Habitat Type interaction (L x H). A p-value of 0 indicates that probability was less than 0.001

	Location main effect					Habitat type main effect									Contrast p-values			L × H
	Galveston Island		Pierce marsh		ANOVA	PREREF		POSTREF		Terrace pond		Terrace marsh		ANOVA				
	n=31		n=31			n=20		n=14		n=14		n=14						
	Mean	S. E.	Mean	S. E.	p-value	Mean	S. E.	Mean	S. E.	Mean	S. E.	p-value	S. E.	p-value	C-1	C-2	C-3	p-value
Spring																		
Salinity	28.81	0.29	25.26	0.16	0	26.75	0.19	27.14	0.59	27.21	0.82	0.07	0.73	0.9204				0.07
Turbidity (NTU)	10.40	0.91	73.96	14.04	0	20.95	2.16	48.19	18.80	25.32	5.22	0.01	25.59	0.0104	0.0658	0.1702	0.033	0.01
Depth (cm)	77.70	3.76	44.11	3.13	0	74.15	3.89	69.63	6.73	62.37	5.63	0.59	4.43	0	0.13986	0.042	0	0.59
Temperature (C°)	25.93	0.23	26.67	0.28	0.014	25.16	0.09	25.84	0.37	27.04	0.38	0	0.28	0.0428	0.13986	0.042	0	0
Dissolved oxygen (ppm)	6.40	0.18	6.57	0.08	0.623	6.09	0.16	6.66	0.20	6.56	0.20	0	0.21	0.0296	0.132	0.001	0	0
Stems (m^2)	40.10	16.12	40.84	15.74	0.83	0	0	0	0	0	0	0.98	19.72	0	1	1	0	0.98
Distance to edge (m)	46.20	10.55	88.73	17.85	0.105	115.25	21.92	115.79	19.50	17.37	2.01	0.28	0.11	0	1	0	0	0.28
Fall																		
Salinity	14.16	0.33	18.81	1.07	0.968	21.60	1.31	13.79	0.41	14.00	0.31	0	0.31	0	0	0.5898	0.165	0
Turbidity (NTU)	26.00	2.47	20.86	1.28	0.09	32.48	3.11	19.17	1.29	19.25	1.21	0	2.29	0.0104	0	1	0.926	0
Depth (cm)	58.23	4.13	53.62	3.62	0.211	68.20	2.04	68.13	4.07	61.11	2.32	0.3	1.22	0	1	0.022	0	0.3
Temperature (C°)	26.05	0.53	23.35	0.89	0.002	26.15	0.94	24.93	0.93	23.68	1.26	0	1.15	0.2322				0
Dissolved oxygen (ppm)	6.41	0.15	7.45	0.19	0	5.89	0.11	7.57	0.25	7.28	0.26	0.49	0.22	0	0	0.178	0.548	0.49
Stems (m^2)	24.39	8.58	24.87	9.02	0.968	0	0	0	0	0	0	1	9.13	0				1
Distance to edge (m)	40.31	10.37	109.84	31.60	0.046	132.10	43.05	124.16	31.31	18.84	1.55	0	0.06	0.0056	0.8508	0.0216	0.008	0

species are associated with shallow open water habitat (Minello 1999).

Seagrasses occurred in the project area during the recent past (Minello 1999) and have re-established in recent years (King and Sheridan 2006; Rozas et al. 2007). Tropical Storm Frances came ashore at Galveston Island September 10, 1998 after the spring sampling date, but a month before we collected the fall PREREF samples. Turbidity and salinity were notably higher and DO lower in fall at PREREF sample sites than an 11-year average for the area reported by Rozas et al. (2007). The effect of the 1998 tropical storm on the water quality or any number of climatic and microhabitat conditions may have led to the failure of the 1999 seagrass planting effort. During our study, seagrass beds were present near GI in Dana Cove, located approximately 2 km east of the GI terraces, and this habitat was examined by King and Sheridan (2006). These seagrass beds supported higher densities of nekton than open water, and the nekton densities in these seagrass beds were similar to those found in both older stands of seagrass and flooded marsh vegetation (King and Sheridan 2006).

At different locations, and within only 2 years of construction, the marsh terraces supported higher density and biomass of important fishery species (e.g., penaeid shrimps, blue crab) and foraging organisms (e.g., grass shrimp) than the habitat type they replaced. Our study included two locations that differed environmentally and ecologically. Marsh terracing improved fishery habitat at both terrace locations, despite these differences. While we confirm that marsh terracing can be an effective tool for enhancing fishery habitat and restoring degraded coastal systems, future studies should use additional metrics (e.g., growth, survival, production) to assess restoration performance.

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